Appendix A. Profile of the Covered Species

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A.1 DESCRIPTION OF THE COVERED SPECIES

A.1.1 Chinook Salmon (*Oncorhynchus tshawytscha*)

A.1.1.1 Listing Status

On March 14, April 4, and May 23, 1994, NMFS received petitions to list several populations of salmon comprising four biological species of Pacific salmon, including chinook salmon, and subsequently initiated comprehensive coastwide status reviews to determine if listings were warranted (September 12, 1994, 59 FR 46808). On February 1, 1995, NMFS was again petitioned to list chinook salmon throughout its range in California, Oregon, Washington and Idaho and again initiated a status review to determine if the petitioned action was warranted (June 8, 1995, 60 FR 30263). On March 9, 1998 (63 FR 11482), NMFS proposed to list the Southern Oregon and California Coastal chinook salmon ESU as threatened. This ESU includes all naturally spawned coastal spring and fall run chinook salmon spawning from Cape Blanco (inclusive of the Elk River) to the southern extent of the current range for chinook salmon at Point Bonita (the northern land mass marking the entrance to San Francisco Bay).

A.1.1.1.1 <u>California Coastal Chinook Salmon ESU</u>

On September 16, 1999 (64 FR 50394), NMFS determined that new information supports a threatened listing for a revised California Coastal chinook salmon ESU, that was part of the larger Southern Oregon and California Coastal chinook salmon ESU, This ESU consists of California coastal chinook salmon populations from Redwood Creek in Humboldt County south through the Russian River in Sonoma County.

Critical habitat for this ESU is designated to include all river reaches and estuarine areas accessible to listed chinook salmon from Redwood Creek (Humboldt County, California) to the Russian River (Sonoma County, California), inclusive (February 16, 2000, 65 FR 7764). Rivers, estuaries, and bays known to support California Coastal chinook salmon include Humboldt Bay, Redwood Creek, and the Mad, Eel, Mattole, and Russian Rivers. Also included are adjacent riparian zones. Excluded are tribal lands and areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years). Major river basins containing spawning and rearing habitat for this ESU comprise approximately 8,061 square miles in California. The following counties lie partially or wholly within these basins (or contain migration habitat for the species): Glenn, Humboldt, Lake, Marin, Mendocino, Sonoma, and Trinity.

A.1.1.1.2 <u>Southern Oregon and Northern California Coastal Chinook Salmon ESU</u>

The Southern Oregon and Northern California Coastal chinook salmon ESU was determined not to warrant listing (September 16, 1999, 64 FR 50394). It includes all naturally spawned populations of chinook salmon from rivers and streams between Cape Blanco, Oregon (excluding the Elk River), and the lower Klamath River, California, excluding populations in the Klamath River Basin upstream from the confluence of the Klamath and Trinity Rivers. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 6,528 square miles in California and Oregon. The following counties lie partially or wholly within these basins: California - Del Norte, Humboldt, and Siskiyou; Oregon - Coos, Curry, Douglas, Jackson, Josephine, and Klamath.

A.1.1.1.3 *Upper Klamath and Trinity River Chinook Salmon ESU*

The Upper Klamath and Trinity River Chinook salmon ESU was determined not to warrant listing (March 8, 1998, 63 FR 11482). It includes all naturally spawned populations of chinook salmon in the Klamath and Trinity River basins upstream of the confluence of the Klamath and Trinity Rivers. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 6,429 square miles in California. The following counties lie partially or wholly within these basins: Del Norte, Humboldt, Siskiyou and Trinity.

A.1.1.2 Status of ESU Populations

A.1.1.2.1 *Southern Oregon and Northern California Coastal Chinook Salmon ESU*

A summary of the status of populations of this ESU are shown in Table A-1. Previous assessments of stocks within this ESU have identified several stocks as being at risk or of concern. Nehlsen et al. (1991) identified seven stocks as at high extinction risk and seven stocks as at moderate extinction risk. Higgins et al. (1992) provided a more detailed analysis of some of these stocks, and identified nine chinook salmon stocks as at risk or of concern. Four of these stocks agreed with the Nehlsen et al. (1991) designations, while five fall-run chinook salmon stocks were either reassessed from a moderate risk of extinction to stocks of concern (Redwood Creek, Mad River, and Eel River) or were additions to the Nehlsen et al. (1991) list as stocks of special concern (Little and Bear Rivers). In addition, two fall-run stocks (Smith and Russian Rivers) that Nehlsen et al. (1991) listed as at moderate extinction risk were deleted from the list of stocks at risk by Higgins et al. (1992), although the USFWS (1997a as cited by NMFS, 1998) reported that the deletion for the Russian River was due to a finding that the stock was extinct. Nickelson et al. (1992) considered 11 chinook salmon stocks within the ESU, of which 4 (Applegate River fall run, Middle and Upper Rogue River fall runs, and Upper Rogue River spring run) were identified as healthy, 6 as depressed, and 1 (Chetco River fall run) as of special concern due to hatchery strays. Huntington et al. (1996 as cited by NMFS 1998) identified three healthy Level II fall-run stocks in their survey (Applegate and Middle and Upper Rogue Rivers).

No current information was available for many river systems in the southern portion of this ESU, which historically maintained numerous large populations. These populations form a genetically distinct subgroup within the ESU. NMFS concluded these California coastal populations do not form a separate ESU. However, they represent a considerable portion of genetic and ecological diversity within this ESU.

Current hatchery contribution to overall abundance is relatively low except for the Rogue River spring run, which also contains almost all of the documented spring-run abundance in this ESU. Fall-run chinook salmon in the Rogue River represent the only relatively healthy population NMFS could identify in this ESU. And found it questionable whether there are sustainable populations outside the Rogue River Basin. All river basins have degraded habitats resulting from agricultural and forestry practices, water diversions, urbanization, mining, and severe recent flooding.

Table A-1. Status of Southern Oregon and Northern California Coastal Chinook Salmon ESU (NMFS 1998).

				Status summaries[3]							Recent abundance		Trends		
River Basin	Sub-basin	Run[1]	Production[2]	A	B	C	D	E	P?[4]	Data Years	Data Type[5]	5-Year Geo. Mean[6]	Long-term[7]	Short-term[8]	Data References
Hunter Creek		Fa		A		D			P						
	Upper	Fa	Natural							1986-96	PI		36.3	36.3	BE and LGL 1995, ODFW 1997e, PSMFC 1997b
Winchuck R	Bear Cr	Fa	Natural	B		D			P	1964-96	AC/PI	592	-2.3	12.0	Nicholas and Hankin 1988, ODFW 1993, BE and LGL 1995, PFMC 1997, PSMFC 1997b
Smith R		Sp		A	A										
		Fa		B					P						
	South Fork	Sp							P	1991-97	SC		30.7	+30.7 (1987-97)	USFS 1997a
	Middle Fork	Sp							P	1991-97	SC		-4.4	-4.4 (1987-97)	USFS 1997a
	North Fork	Sp							P	1992-96	SC		26.2		USFS 1997a
	Mill Cr	Fa	Mixed							1980-96	SC		-1.1	1.9	BE and LGL 1995, PSMFC 1997b, Waldvogel 1997
Klamath R	Lower tributaries	Fa		B	B				P						
	Blue Cr	Fa								1988-96	SN		14.9	14.9	YTFP 1997b
Redwood Cr		Fa		B	C				P						
	Little R	Fa			C				P						
Mad R		Fa		B	C				P						
	North Fork	Fa	Mixed							1985-93	SC		-29.0		BE and LGL 1995, PSMFC 1997b
	Canon Cr	Fa	Natural					<TD DTH ="3 %"		1964-97	PI		-4.9	+0.1 (1987-97)	PFMC 1997
Humboldt Bay	Tributaries	Fa		A	A				P						
Eel R		Fa			C				P	1951-97	DC	16	3.6	-29.7 (1987-97)	PSMFC 1997b, SEC 1997
	Lower	Fa		B											
	Sprowl Cr	Fa	Natural							1967-97	PI		-4.7	-12.4 (1987-97)	PFMC 1997
	Tomki Cr	Fa	Natural							1964-97	TE	25	-15.6	-37.5 (1987-97)	BE and LGL 1995, PFMC 1997, PSMFC 1997b
	South Fork	Fa	Natural							1938-75	WC	4,022	-0.2		BE and LGL 1995

Table A-1. Status of Southern Oregon and Northern California Coastal Chinook Salmon ESU (NMFS 1998) (Continued)

NOTES

1 Run timing designations: Fa -- fall; Sp -- spring; Su -- summer; Wi -- winter (as reported by data reference).

2 Production: (as reported by data reference).

3 Status summaries from the following sources:

A--Nehlsen et al. (1991):

E, endangered (US); X, extinct; A+, possibly extinct; A, high extinction risk; B, moderate extinction risk; C, special concern.

B--Higgins et al. (1992):

A, high risk of extinction; B, moderate risk of extinction; C, stock of concern.

C--Nickelson et al. (1992):

H, healthy; D, depressed; S, special concern; U, unknown.

1, May not be a viable population; 2, Hatchery strays; 3, Small, variable run.

D--WDF et al. (1993): Three characters represent stock origin, production type, and status, in that order.

Origin: N, native; M, mixed; X, non-native; U, unknown; -, unresolved by state and tribes.

Production: W, wild; C, composite; A, cultured; U, unknown; -, unresolved.

Status: H, healthy; D, depressed; C, critical; U, unknown.

E--Huntington et al. (1996):

H-I, healthy Level I (abundance at least two-thirds as great as would be found in the absense of human impacts).

H-II, healthy Level II (abundance between one-third and two thirds as great as expected without human impacts).

4 Petition status [P?]: Indicates (by 'P') stocks included in the ONRC and Nawa petition dated 31 January 1995. Parentheses indicate stock is included as part of a larger unit in the petition.

5 Data Type Codes:

AC, angler catch expanded (1988-92); CS, carcass; DC, dam count; FM, fish per mile; HE, total estimated hatchery escapement; IT, index total; PC, peak or index live fish, surveys combined; PI, peak or index live fish; PR, peak redd count; RC, redd count; RH, resting hole counts; RM, redds per mile; RMC, redds per mile (surveys combined); SC, spawner counts; SN, snorkle counts; TC, trap count; TE, total estimated escapement (includes hatchery escapement only for mixed production type); TL, total live fish count; WC, wier count.

6 Most recent 5 years of data used to calculate spawning escapement geometric mean. (Expanded angler catch = 1988-92).

7 Trend (Long-term): Calculated for all data collected after 1950.

8 Short-term Trend: Calculated for most recent 7-10 years during the period 1987-96, except as noted.

NMFS was very concerned about the risks to spring-run chinook in this ESU; their stocks are in low abundance and they have continued to decline dramatically in recent years. In addition, the lack of population monitoring, particularly in the California portion of the range, led to a high degree of uncertainty regarding the status of these populations.

NMFS (1998) concluded that chinook salmon in this ESU are likely to become at risk of extinction in the foreseeable future. Overall abundance of spawners is highly variable among populations, with populations in California and spring-run chinook salmon throughout the ESU being of particular concern. There is a general pattern of downward trends in abundance in most populations for which data are available, with declines being especially pronounced in spring-run populations. NMFS found that extremely depressed status of almost all coastal populations south of the Klamath River is an important source of risk to the ESU.

A.1.1.3 Distribution

Native spawning populations of chinook salmon are distributed along the Asian coast from Hokkaido, Japan to the Anadyr River and along the North American coast from central California to Kotzebue, Alaska (Moyle 1976; Allen and Hassler 1986; Healey 1991). Chinook salmon spawning may occur from near tidewater in coastal watersheds to over 3,200 km upstream in headwaters of the Yukon River (Major et al. 1978). Introductions of juvenile chinook salmon have also established naturally reproducing populations in New Zealand, Chile and the Great Lakes.

A.1.1.4 Life History

The variable life history patterns of chinook salmon have been thoroughly reviewed by Allen and Hassler (1986) and Healey (1991). Healey (1991) presented a conceptual model that summarized two main components of variation within chinook salmon life histories. The first component is racial, which accounts for the two main behavioral types. "Stream-type" chinook typically spend one or more years as juveniles in fresh water, undertake extensive salt water migrations and return to natal watersheds in the spring or summer several months prior to spawning (Healey 1991). Stream-type chinook are typical of Asian populations and of northern populations and headwater tributaries of southern populations in North America. "Ocean-type" chinook generally migrate to the ocean within three months after emergence, stay within coastal waters during their ocean phase and return to natal watersheds in the fall several days or weeks prior to spawning (Healey 1991). Ocean-type chinook are typical of populations along the North American coast south of 56^{o} N (Healey 1991).

The second component of the life history model is tactical and accounts for variation within each race (Healey 1991). Chinook salmon populations have evolved a range of juvenile and adult behavior patterns that spreads risk across years and across habitats. These patterns include variations in timing of juvenile migrations, variations in length of estuarine residency, variations in age of maturity and variations in adult run timing (Allen and Hassler 1986; Healey 1991).

Chinook salmon in California return to spawn at two to seven years of age, with three and four year olds comprising the bulk of spawning populations. Two year old males are called jacks or grilse and may comprise 10% to 25% of a spawning run (Allen and

Hassler 1986). Spring runs of chinook (stream-type) generally enter watersheds in May and June, but will not spawn until September and October. The chinook population in the Klamath River is predominately a late August/September run, with spawning occurring from October through December (Snyder 1931; Allen and Hassler 1986). The timing of fall runs (ocean-type) in coastal watersheds is variable and highly influenced by rainfall and stream discharge. Sand bars at the mouths of coastal watersheds must often breach before chinook salmon can enter. Runs may occur from October through January, depending on rainfall.

The fecundity of female chinook salmon is variable, depending on the age and size of the fish and geographic location. Estimates range from 2,000 to 14,000 eggs (Moyle 1976). Klamath River chinook average 3,600 eggs, while Sacramento River fish average 7,300 eggs (Allen and Hassler 1986). After completing her redd female chinook may defend the redd site from four to 25 days, depending on her condition (Neilson and Geen 1981, Neilson and Banford 1983). All chinook salmon eventually die after spawning.The incubation of chinook salmon eggs is inversely related to water temperature. Eggs in 16° C water hatch in about 32 days (Healey 1991). Chinook alevins then spend two to four weeks in the gravel prior to emergence. Survival to emergence is variable and influenced by numerous environmental factors.

A.1.1.5 Habitat Requirements

A.1.1.5.1 Spawning Habitat

Redd sites are selected by female chinook salmon and are usually in pool tails with adequate flow, depth and substrate (Briggs 1953; Allen and Hassler 1986). Velocities of 0.15 to 1.89 m/sec have been recorded at chinook redds (Briggs 1953; Smith 1973; Chapman et al. 1986). Riffle depths at redd sites may range from five to 700 cm (Chapman et al. 1986, Healey 1991). Typically, spring and fall run chinook spawn in 30 to 120 cm of water (Chapman 1949). Chinook salmon construct redds in gravels ranging from 1.3 to 10.2 cm in diameter (Allen and Hassler 1986). Eggs are usually buried 20 to 60 cm below the surface of a completed redd (Briggs 1953). The requirement of sufficient subgravel water flow seems to be of more importance to chinook salmon spawning success relative to other salmonid species (Healey 1991). Chinook produce the largest eggs which have the smallest surface area -to-volume ratio of all salmonid species. Healey (1991) speculates that chinook eggs would be more sensitive to reduced oxygen levels and require a more certain rate subgravel water flow.

A.1.1.5.2 Rearing Habitat

A large downstream migration of chinook fry right after emergence is common in most populations, and may be a dispersal mechanism to distribute fry among all suitable rearing habitats (Bjornn 1971; Reimers 1971). Once started downstream, chinook fry may continue to the estuary or take up residence in the watershed for a period ranging from several weeks to a year or more (Healey 1991). Residing fry will initially seek cover along channel margins or in low velocity areas associated with the channel bottom. As they grow larger they tend to establish territories in faster, deeper habitats (Everest and Chapman 1972). Overwintering (stream-type) juveniles seek shelter under large boulders and woody debris, a habitat shift probably caused by lower water temperatures and increased flows (Chapman and Bjornn 1969).

Estuaries play a vital role in the life cycle of chinook salmon. Fry of ocean-type chinook often migrate downstream immediately after emergence and rear to smolt size in estuaries (Healey 1991). Chinook migrating as young-of-the-year or yearling smolts also rely on estuarine habitat for additional growth and acclimation to saline water prior to oceanic migrations. There is a tendency for ocean-type chinook juveniles to make extensive use of estuarine habitat, whereas stream-type chinook juveniles briefly utilize their watershed's estuary (Healey 1991).

A.1.2 Coho Salmon (*Oncorhynchus kisutch*)

A.1.2.1 Listing Status

On October 20, 1993, the National Marine Fisheries Service (NMFS) received a petition to list coho salmon throughout its range in Washington, Oregon, Idaho, and California, and subsequently initiated a status review to determine if the petitioned action was warranted (January 26, 1994, 59 FR 3662). On July 25, 1995 (60 FR 38011), NMFS published a proposed rule to list the Southern Oregon/Northern California Coasts (SONCC) coho salmon evolutionarily significant unit [1] (ESU) as threatened. This ESU extends from Cape Blanco in Curry County, Oregon, to Punta Gorda in Humboldt County, California. On May 6, 1997 (62 FR 24588), NMFS listed the SONCC coho salmon ESU as threatened.

On November 25, 1997 (62 FR 62741), NMFS published a proposed rule to designate critical habitat for SONCC coho salmon. Critical habitat for SONCC coho salmon was designated on May 5, 1999 (64 FR 24049) and encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between Cape Blanco and Punta Gorda. Excluded are areas above specific dams or above longstanding, naturally impassable barriers (i.e., natural waterfalls in existence for at least several hundred years). Major river basins containing spawning and rearing habitat for this ESU comprise approximately 18,090 square miles in California and Oregon. The following counties lie partially or wholly within watersheds inhabited by this ESU: California - Del Norte, Glenn, Humboldt, Lake, Mendocino, Siskiyou, and Trinity; Oregon - Coos, Curry, Douglas, Jackson, Josephine, and Klamath. More detailed critical habitat information (i.e., specific watersheds, migration barriers, habitat features, and special management considerations) for this ESU can be found in 64 FR 24049.

A.1.2.2 Status of ESU Populations

Risk to Populations of Southern Oregon/Northern California Coasts ESU (from: NOAA-NWFSC Tech Memo-24: Status Review of Coho Salmon; (NMFS, 1994a)

All coho salmon stocks between Punta Gorda and Cape Blanco are depressed relative to past abundance, but there are limited data to assess population numbers or trends. The main stocks in this region (Rogue River, Klamath River, and Trinity River) are heavily influenced by hatcheries and, apparently, have little natural production in mainstem rivers. The apparent declines in production in these rivers, in conjunction with

[1] An Evolutionarily Significant Unit is a distinct population segment that is substantially reproductively isolated from other conspecific population units and represents an important component in the evolutionary legacy of the species (Waples 1991).

heavy hatchery production, suggest that the natural populations are not self-sustaining. The status of coho salmon stocks in most small coastal tributaries is not well known, but these populations are small. There was unanimous agreement among the Biological Review Team (BRT) that coho salmon in this ESU are not in danger of extinction but are likely to become endangered in the foreseeable future if present trends continue (Table A-2).

Table A-2. Summary of risk considerations for Southern Oregon/Northern California Coasts ESU (NMFS 1994a).

Risk category	Considerations
Absolute numbers (Recent average)	Run size ca. 10,000 natural, 20,000 hatchery. Current production largely in the Rogue and Klamath basins.
Numbers relative to historical abundance and carrying capacity	Substantially below historical levels. In California portion of ESU, ca. 36% of coho streams no longer have spawning runs. Widespread habitat degradation.
Trends in abundance and production	Long-term trends clearly downward. Main data are for Rogue River basin, where runs declined to very low levels in 1960s and 1970s, then increased with start of hatchery production.
Variability factors	Low abundance or degraded habitat may increase variability.
Threats to genetic integrity	Most existing populations have hatchery plantings, with many out-of-state stock transfers in California portion of the ESU.
Recent events	Recent droughts and change in ocean production have probably reduced run sizes.
Other Factors	None identified.
Conclusion	Not presently in danger of extinction, but likely to become so.

A.1.2.3 Distribution

Globally, coho salmon spawn in coastal watersheds in both Asia and North America. In Asia they are distributed from Hokkaido, Japan to the Anadyr River in Russian Siberia (Moyle 1976; Hassler 1987). In North America coho salmon are distributed from Point Hope, Alaska south to the northern edge of Monterey Bay (Moyle 1976). Along the North American coast coho salmon are most abundant between southern Oregon and southeast Alaska. In California, coho salmon are the second most abundant of the five species of Pacific salmon. They are found in numerous coastal drainages from the Oregon/California border to Waddel and San Lorenzo Creeks of Santa Cruz county (Sandercock 1991). Coho salmon are uncommon and, in the Sacramento River despite several attempts (1956, 1957 and 1958) to establish populations through plantings of juveniles (Hallock and Fry 1967).

The Southern Oregon/northern California coasts coho ESU includes coho salmon from Cape Blanco in southern Oregon to Punta Gorda in northern California. Geologically, this region includes the Klamath Mountains Geologic Province, which has soils that are not as erosive as those of the Franciscan Formation to the south (NMFS, 1994a). Dominant vegetation along the coast is redwood forest, while some interior basins are much drier than surrounding areas and are characterized by many endemic plant species. Elevated stream temperatures are a factor in some areas, but not to the extent that they are in areas south of Punta Gorda.

Rivers in this ESU are relatively long compared to those to the south. With the exception of major river basins such as the Rogue and Klamath, most streams in this region have short duration of peak flows and relatively low flows given both peak flow levels and basin sizes, compared to rivers farther north (NMFS 1994a). Freshwater fishes include elements of the Sacramento River fauna as well as from the Klamath-Rogue ichthyofaunal region. Strong and consistent coastal upwelling begins around Cape Blanco and continues south into central California, resulting in a relatively productive nearshore marine environment. In contrast to coho salmon from north of Cape Blanco, which are most frequently captured off the Oregon coast, coho salmon from this region are captured primarily in California waters.

Genetic data indicate that most samples from this region differ substantially from coho salmon from south of Punta Gorda. In general, populations from southern Oregon also differ from coastal Oregon populations north of Cape Blanco. However, some samples from the Rogue River show an unexplained genetic affinity to samples from outside the region, including some from the Columbia River. In addition, a sample from the Elk River (just south of Cape Blanco) clustered with samples from the Umpqua River (NMFS 1994a).

The southern boundary of this ESU is farther south than the boundary designated for the Klamath Mountains Province steelhead ESU, which includes the Klamath River but not drainages to the south (Busby et al. 1994 as cited by NMFS 1994a). Both the steelhead and coho salmon ESUs share the northern boundary of Cape Blanco. Although the Klamath River (inclusive) serves as the southern boundary for the Klamath Mountains Geological Province and for freshwater fish faunas, major changes in ocean currents and environmental characteristics, as well as the southern limit of the steelhead half-pounder life history strategy, occur at Cape Mendocino/Punta Gorda.

Consequently, the southern limit of the steelhead ESU was based primarily on strong genetic discontinuity between Klamath River steelhead and steelhead populations to the south (Busby et al. 1994 as cited by NMFS 1994a). In contrast, Punta Gorda serves as the southern boundary of the southern Oregon/northern California coho salmon ESU because of the strong environmental transition at Punta Gorda, and because genetic data indicate Punta Gorda, rather than the Klamath River, as an approximate transition area for coho salmon.

For California coho salmon, Pacific Rivers Council et al. (1993 as cited by NMFS, 1994a) reported that Brown and Moyle (1991) estimated that naturally spawned adult coho salmon (regardless of origin) returning to California streams were less than 1% of their abundance at mid-century, and indigenous, wild coho salmon populations in California did not exceed 100 to 1,300 individuals. They further state that Brown and Moyle (1991) found that 46% of California streams, which historically supported coho salmon populations, and for which recent data were available, no longer supported runs (NMFS 1994a).

A.1.2.4 Life History

The life history of coho salmon in California has been well documented by several authors (Shapovalov and Taft 1954; Moyle 1976; Hassler 1987; Sandercock 1991). The life cycle of coho salmon is from two to five years, with three years being most common. Juveniles usually spend at least one year in freshwater before out-migrating to the ocean (juveniles in Alaskan watersheds commonly reside for two years). Coho salmon from California watersheds then spend one to two years at sea before returning to spawn in their natal watersheds (Alaskan coho may stay at sea for three years). The primary exception to this pattern are jacks, sexually mature males that return to freshwater to spawn after only 5-7 months in the ocean (NMFS 1994a). Jacks are a highly variable component of a spawning run. For example, Murphy (1952) summarized counts of coho salmon passing over Benbow Dam on the South Fork of the Eel River from 1939-51 and jacks comprised from 6.9% to 33.8% of the total coho escapement. There is a latitudinal cline in the proportion of jacks in a coho salmon population, with populations in California having more jacks and those in British Columbia having almost none (Drucker 1972 as cited by NMFS 1994a). Although the production of jacks is a heritable trait in coho salmon (Iwamoto et al. 1984), it is also strongly influenced by environmental factors (Shapovalov and Taft 1954, Silverstein and Hershberger 1992 as cited by NMFS 1994a). The proportion of jacks in a given coho salmon population appears to be highly variable and may range from less than 6% to over 43% over 9-35 years of monitoring (Shapovalov and Taft 1954, Fraser et al. 1983, Cramer and Cramer 1994 as cited by NMFS 1994a).

Spawning occurs from early September through March, with peak periods between November and January. In the Klamath River, returning coho enter between September and December, with most spawning occurring during October and November. However, many spawning runs in California occur only after heavy rains have elevated stream flows to breach sand bars at the mouths of some coastal watersheds. If conditions (flow, temperature) in a coastal watershed are unsuitable, coho will postpone migration for weeks or months until conditions change (Sandercock 1991). Coho in large watersheds such as the Klamath River may migrate 65 to 130 miles to spawning sites in tributaries. Coho in smaller coastal watersheds rarely migrate more than 60 miles before spawning in upper sections of main channels or in smaller tributaries. There is also a tendency of earlier run fish to migrate further upstream than late run fish (Briggs 1953). After completing her redd, female coho salmon may remain near the redd for three to 23 days and defend the redd site until too weak to do so (Briggs 1953). All coho salmon die after spawning.

Fecundity of female coho salmon is variable depending on size of female, geographic location and age of spawner. Hassler (1987) cited values of 1,440 to 5,700 eggs for spawners of 44 to 72 cm from Washington. Shapovalov and Taft (1957) reported an average fecundity of 2,700 eggs from Waddell and San Lozenzo Creeks. Ocean distribution of coho salmon, inferred from marine recoveries of coded-wire-tagged fish, show distinctive differences between regions. Coded-wire tags (CWTs) are primarily recovered in salt or fresh water as the salmon return to their natal streams after overwintering in the ocean (NMFS 1994a). Ocean distribution patterns based on CWT marine recovery patterns have been determined from CWT recovery data for 66 North American hatcheries from the Pacific States Marine Fisheries Commission's (PSMFC 1994 as cited by NMFS 1994a). Ocean distribution patterns for California coho salmon are shown in Table A-3.

Table A-3. Marine recoveries of coded wire tags, expanded for sampling, from selected production facilities in Alaska (AK), British Columbia (BC), Washington (WA), Oregon (OR), and California (CA) by release location, including years released, expanded number of tags recovered by state or province, total number of tags recovered, and percent recoveries by state or province (Data from PSMFC 1994 as cited by NMFS 1994a).

Hatchery	Brood years	Expanded number of marine recoveries (% of total)					
		AK	BC	WA	OR	CA	Total
Iron Gate	1974, 77-84, 88-89	0.0 (0.0)	6.4 (0.1)	14.5 (0.2)	1,715.6 (19.4)	7,098.5 (80.3)	8,835.0
Trinity River	1976-85, 89	0.0 (0.0)	4.0 (0.0)	27.5 (0.1)	4,610.5 (22.5)	15,820.5 (77.3)	20,462.5
Mad River	1975, 78-79, 84-86	0.0 (0.0)	1.1 (0.0)	16.3 (0.7)	495.2 (20.2)	1,933.1 (79.0)	2,445.7
Warm Springs	1984-87	0.0 (0.0)	0.0 (0.0)	2.7 (0.3)	59.9 (7.2)	764.0 (92.4)	826.6

The patterns of recoveries showed marked differences between areas, with extremely limited transition zones between areas (NMFS, 1994a). Eight general CWT recovery patterns were identified, one of which includes Northern California and Oregon south of Cape Blanco. Coho salmon released from the southernmost facilities (those south of Cape Blanco) had the most southerly recovery patterns: these fish were recovered primarily in California (65-92%), with some recoveries in Oregon (7-34%) and almost none (<1%) in Washington or British Columbia. The recovery pattern of coho salmon released from the southernmost hatchery, Warm Springs (Russian River), had a much higher proportion of California recoveries (92%) than the other California and southern Oregon facilities. Whether this represents a unique recovery pattern, or results from the southerly location of the hatchery, is not known (NMFS 1994a).

A.1.2.5 Habitat Requirements

A.1.2.5.1 Spawning Habitat

Redd sites are selected by females and are located in pool tails or slightly upstream of the hydraulic control, where the water changes from a laminar to more turbulent flow. Water depths at redd locations range from 0.18 to 0.46 meters (Smith 1973; Hassler 1987). Redds are located in relatively fast water (0.3 to 0.5 m/sec) which ensures adequate aeration and circulation to facilitate embryo development and fry emergence (Smith 1973; Hassler 1987). Coho salmon utilize small to medium sized substrate ranging from 1.3 to 15.0 cm in diameter (Reiser and Bjornn 1979; Sandercock 1991). Developing coho salmon appear able to tolerate higher concentrations of fines (up to 10%) than other salmonid species, although redds situated in gravels with lower amounts of fines (5% or less) have higher rates of juvenile emergence (Emmett et al. 1991). Excessive amounts of fines deposited on redds reduces oxygen flow to developing eggs and young and impedes successful emergence of juveniles. Briggs

(1953) reported that coho salmon in California spawn in water temperatures ranging from 5.6° C to 13.3° C.

Incubation of eggs takes from 38 to 101 days and is inversely related to water temperature (Hassler 1987). Egg development is slower in colder water and faster in warmer water. After hatching, coho alevins remain in the gravel until their yolk sacs have been absorbed, usually a period of two to ten weeks (Moyle 1976; Hassler 1987). Survival of eggs and alevins to emergence is highly variable and dependent on numerous environmental factors. Under extreme conditions none may survive; under average conditions 15%-27% may survive (Neave 1949); and under ideal conditions 65%-85% may survive (Shapovalov and Taft 1954).

A.1.2.5.2 *<u>Rearing Habitat</u>*

Newly emerged coho fry seek out shallow water along stream margins, backwaters and side channels (Sandercock 1991). Initially coho fry form schools, but as they grow larger the schools break up and juveniles (parr) tend to establish individual territories (Hassler 1987). Larger, more dominant parr tend to occupy the heads of pools; while smaller parr are found farther downstream (Chapman and Bjornn 1969). As the parr grow, their territories expand until by summer they are located in deep pools. Ideal rearing habitat consists of a mixture of pools and riffles with abundant instream and overhead cover (especially large woody debris), water temperatures between 10° and 15° C, dissolved oxygen near saturation and low amounts of fines (Hassler 1987). Scrivener and Andersen (1984) reported that streams with larger amounts of complex habitat (cobbles, boulders, logs and overhanging riparian vegetation) supported larger numbers of juvenile coho salmon.

By the onset of autumn coho parr decrease feeding activity and migrate into deeper pools with LWD and undercut banks, seeking protection from elevated flows. In some watersheds coho parr will move into tributaries that maintain more stable flows throughout the winter (Tripp and McCart 1983). Towards the end of March coho parr start to migrate downstream and into the ocean. In California, out-migration from small coastal watersheds peaks from mid-April to mid-May (Shapovalov and Taft 1954). Factors affecting time of out-migration include: size of juveniles, flow conditions, water temperature, dissolved oxygen, day length and food availability (Shapovalov and Taft 1954). At the onset of out-migration, juveniles defend territories less aggressively and form aggregations. Out-migrants move in groups of 10 to 50 fish and are of similar size (Shapovalov and Taft 1954). Parr marks are still obvious on early migrants, but later migrants are more silvery, having transformed into smolts. Size of coho smolts seems to be consistent throughout the species geographic range. Several authors have reported an average fork length of 10 to 12 cm for coho smolts (Sumner 1953; Shapovalov and Taft 1954; Salo and Bayliff 1958).

A.1.3 Steelhead and Resident Rainbow Trout (*Oncorhynchus mykiss irideus*)

A.1.3.1 Listing Status

A.1.3.1.1 Steelhead

Steelhead from the Illinois River, a Rogue River tributary, were initially petitioned for listing on 5/5/92. On 7/31/92, NMFS published in the *Federal Register* that the listing may be warranted. On May 29, 1993 (58 FR 29390), NMFS concluded that Illinois River winter steelhead did not constitute a "species," and therefore, did not qualify for listing under the ESA. However, NMFS requested biological information for all coastal steelhead populations. On February 16, 1994, NMFS received a petition to list steelhead throughout its range in Washington, Oregon, Idaho, and California, and subsequently initiated a status review to determine if the petitioned action was warranted (May 27, 1994, 59 FR 27527).

Klamath Mountains Province Steelhead

On March 16, 1995 (60 FR 14253), NMFS published a proposed rule to list steelhead in the Klamath Mountains Province (KMP) ESU as threatened. The KMP steelhead ESU was proposed for listing again on August 9, 1996 (61 FR 41541). The KMP steelhead ESU extends from Cape Blanco, Oregon, to the Klamath River Basin, California, inclusive. On March 19, 1998 (63 FR 13347), NMFS determined that listing was not warranted for this ESU. The ESU was reclassified as a candidate for listing due to concerns over specific risk factors, but it was again determined that listing was not warranted for this ESU (66 FR 17845).

Northern California Steelhead

On August 9, 1996 (61 FR 41541), NMFS published a proposed rule to list the Northern California steelhead ESU as threatened. The ESU includes steelhead in California coastal river basins from Redwood Creek south to the Gualala River, inclusive. As with KMP steelhead, on March 19, 1998 (63 FR 13347), NMFS determined that listing was not warranted for the Northern California steelhead ESU. However, the ESU was reclassified as a candidate for listing due to concerns over specific risk factors. Because the State of California has failed to implement conservation measures that NMFS considered critically important in its decision not to list the Northern California steelhead ESU, NMFS completed an updated status review and has reconsidered the status of this ESU under the ESA. On February 11, 2000 (65 FR 6960), NMFS proposed to list Northern California steelhead as threatened. The Northern California steelhead ESU was listed as threatened on June 7, 2000 (65 FR 36075). Critical habitat has not been proposed or designated for Northern California steelhead.

A.1.3.1.2 Resident Rainbow Trout

USFWS recently asserted jurisdiction over the resident form of the rainbow trout, which is genetically indistinguishable from steelhead.

A.1.3.2 Status of Steelhead ESU Populations

A.1.3.2.1 Klamath Mountains Province Steelhead

(From: NOAA-NMFS Tech Memo-19. Status Review for Klamath Mountains Province Steelhead [NMFS 1994b]).

Historical information for northern California populations of steelhead are scarce, although Snyder (1925 as cited by NMFS1994b) noted that trout (including steelhead) were declining in the Klamath River Basin at that time.

Qualitative evaluations considered recent published assessments by agencies or conservation groups of the status of steelhead stocks from Cape Blanco to the Klamath River Basin (Nehlsen et al. 1991; Nickelson et al. 1992; USFS 1993a,b; McEwan and Jackson in prep. (as cited by NMFS 1994b). Results of these assessments are summarized in Table A-4. NMFS (NMFS 1994b) attempted to distinguish naturally produced fish from hatchery produced fish in compiling these summary statistics. All statistics were based on data for adults that spawn in natural habitat ("naturally spawning fish"). The total of all naturally spawning fish ("total run size") is divided into two components "Hatchery produced" fish are reared as juveniles in a hatchery but return as adults to spawn naturally; "naturally produced" fish are progeny of naturally spawning fish (NMFS 1994b).

Table A-4. Summary of recent qualitative assessments of steelhead abundance for all river basins reviewed. Blanks indicate that a particular run was not evaluated (NMFS 1994b).

River basin	Run-type	Nehlsen risk level[a]	ODFW/CDFG assessment[b]	USFS assessment[c]
Oregon				
Elk River	Winter			Healthy
Euchre Creek	Winter			
Rogue River	Winter		Healthy	Healthy
	Summer	Moderate	Depressed	Depressed
Applegate River	Winter			
	Summer			
Illinois River	Winter	Moderate	Depressed	Depressed
Hunter Creek	Winter			
Pistol River	Winter		Depressed	
Chetco River	Winter		Depressed	Depressed
Winchuck River	Winter		Healthy	Healthy
California				
Smith River	Winter		Healthy	Low abundance
	Summer	High		Depressed
Klamath River	Winter			Low abundance, insufficient information
	Summer	Moderate		Depressed, moderate to high risk
Trinity River	Winter			Stable, depressed
	Summer			Stable, high risk

[a] - Risk of local extinction, as defined in Nehlsen et al. (1991).
[b] - Assessments in state agency documents: Oregon, Nickelson et al. (1992); California, McEwan and Jackson (in prep.).
[c] - General assessments of condition of portions of runs on U.S. Forest Service lands (USFS 1993a,b).

The quantitative and qualitative risk evaluation analyses (NMFS, 1994b) revealed the following:

- Although historical trends in overall abundance within the ESU are not clearly understood, there has been a substantial replacement of natural fish with hatchery produced fish.

- Since about 1970, trends in abundance have been downward in most steelhead populations within the ESU, and a number of populations are considered by various agencies and groups to be at moderate to high risk of extinction.

- Declines in summer steelhead populations are of particular concern.

- Most populations of steelhead within the area experience a substantial infusion of naturally spawning hatchery fish each year. After accounting for the contribution of these hatchery fish, we are unable to identify any steelhead populations that are naturally self-sustaining.

- Total abundance of adult steelhead remains fairly large (above 10,000 individuals) in several river basins within the region, but several basins have natural runs below 1,000 adults per year.

The Klamath Mountains Province steelhead ESU was recently reevaluated by NMFS Biological Review Team (66 FR 17845). They reviewed updated abundance and trend information available for this ESU and concluded that the ESU was not in danger of extinction nor likely to become so in the foreseeable future (66 FR 17845).

A.1.3.2.2 *Northern California Steelhead*

(From: NOAA-NMFS NMFS-NWFSC-27 Status Review for West Coast Steelhead from Washington, Idaho, Oregon, and California [NMFS, 1996]).

NMFS review team concluded that the Northern California steelhead ESU is not presently in danger of extinction, but that it is likely to become endangered in the foreseeable future. Nehlson et al.'s (1991) finding's of risk for extinction for Northern California Steelhead are summarized in Table A-5. below.

Population abundances are very low relative to historical estimates (1930s dam counts), and recent trends are downward in stocks for which data were available, except for two small summer steelhead stocks. Summer steelhead abundance is very low. There is particular concern regarding sedimentation and channel restructuring due to floods, apparently resulting in part from poor land management practices. The abundance of introduced Sacramento squawfish as a predator in the Eel River is also of concern.

Table A-5. Northern California Steelhead stocks identified by Nehlsen et al. (1991) as at some risk of extinction.

Extinct	Possibly extinct	High risk	Moderate risk	Special concern
None	None	Redwood Creek Mad River	Eel River	None

For certain rivers (particularly the Mad River), NMFS is concerned about the influence of hatchery stocks, both in terms of genetic introgression and of potential ecological interactions between introduced stocks and native stocks. They found that there are two major areas of uncertainty. Information on steelhead run sizes throughout the ESU is lacking. Their conclusions were based largely on evidence of habitat degradation and the few dam counts and survey index estimates of stock trends in the region. Also, the genetic heritage of the natural winter steelhead population in the Mad River is uncertain. Table A-6. summarizes the spawning escapement estimates for rivers within the Northern California Coastal Steelhead ESU as of the 1960's. Table A-7 provides additional abundance estimates.

Risk factors identified for this ESU include freshwater habitat deterioration due to sedimentation and flooding related to land management practices and introduced Sacramento squawfish as a predator in the Eel River. For certain rivers (particularly the Mad River), NMFS is concerned about the influence of hatchery stocks, both in terms of genetic introgression and potential ecological interactions between introduced stocks and native stocks.

Table A-6. Estimated steelhead spawning populations for Northern California Steelhead ESU rivers in the mid-1960s (CDFG 1965 as cited in NMFS 1996), with comparable recent maximum estimates.

Stream	Population Estimate
Redwood Creek	10,000
Mad River	6,000
Eel River System (Total)	82,000
Mattole River	12,000
Ten Mile River	9,000
Noyo River	8,000
Big River	12,000
Navarro River	16,000
Garcia River	4,000
Gualala River	16,000
Other streams (Humboldt, Mendocino Counties)	23,000
Total	**198,000**

Table A-7. Summary of historical abundance estimates for the Northern California evolutionarily significant unit (as cited in NMFS 1996).

River Basin	Abundance*	Years	Reference
Eel River			
Cape Horn Dam	4,400	1930s	McEwan and Jackson 1996
	1,000	1980s	McEwan and Jackson 1996
Benbow Dam	18,784	1940s	Shapovalov and Taft 1954
	3,355	1970s	McEwan and Jackson 1996
Mad River			
Sweasy Dam	3,800	1940s	Murphy and Shapovalov 1951
	2,000	1960s	McEwan and Jackson 1996
Casper Creek	114	1964	Graves and Burns 1970
	102	1968	Graves and Burns 1970
* Excludes estimates from CDFG 1965.			

A.1.3.3 Distribution

Steelhead are widely distributed from the Kuskokwin River of western Alaska to Baja California (Moyle 1976; Behnke 1992). The anadromous rainbow trout is called the steelhead, which accounts for most of the variable life history patterns. Steelhead populations occur throughout the range of steelhead except in the northern and southern extremities (Behnke 1992). The present southern limit of steelhead distribution is Malibu Creek, California. The southern range of summer run steelhead is the Middle Fork of the Eel River (Barnhart 1986).

A.1.3.4 Life History

The life histories of rainbow trouthave been reviewed by numerous authors (Smith 1973; Jones 1976; Moyle 1976; Barnhart 1986; Behnke 1992). The anadromous and resident forms are genetically indistinguishable, and the life history of resident rainbow trout are similar to those of steelhead while in the freshwater phase.

Steelhead populations may be grossly categorized as summer run or fall/winter run fish, depending when spawning adults enter fresh water. This is an oversimplification and adult steelhead probably enter fresh water every month of the year somewhere in their widespread distribution (Behnke 1992). Summer run steelhead are not abundant throughout the Pacific southwest and the runs in many watersheds consist of less than 100 adults (Roelofs 1983).

Summer run fish usually enter fresh water from May through August and move upstream to hold in deep pools until the following winter or spring to spawn. These stream-maturing type steelhead enter fresh water in a sexually immature condition and require several months in freshwater to mature prior to spawning Fall/winter run fish generally enter fresh water from September through November, whereas many coastal watersheds have late runs of winter steelhead that enter fresh water from January through April. These ocean-maturing type steelhead enter fresh water with well-developed gonads and spawn shortly after river entry The partitioning of an anadromous species into distinct races is an excellent reproductive strategy since this enlarges the use of its environment and increases productivity (Behnke 1992).

Adult steelhead are iteroparous and can spawn more than once before dying. Repeat spawners are a significant contribution to many populations. Most populations consist of 10% to 20% repeat spawners (Behnke 1992). Forsgren (1979) reported that second time spawners comprise 70% to 85% of repeat spawners and third time spawners comprise 10% to 25% of repeat spawners. Spawning survival is highly variable and influenced by genetic factors, habitat quality, fishing pressure and management plans.

The fecundity of rainbow trout (either resident or anadromous) is highly variable, from 200 to 12,000 eggs depending on the size of the female (Moyle 1976). Moyle (1976) reported that resident fish usually produce less than 1,000 eggs and that steelhead average about 2,000 eggs per kilogram of body weight.

Incubation of steelhead eggs, as with all salmonids, is inversely related to water temperature. Eggs in 15°C water hatch in approximately 19 days, whereas eggs in 5°C hatch in about 80 days (Barnhart 1986). Steelhead alevins remain in the gravel for two to four weeks and are sustained by their yolk sacs. Survival of eggs and alevins to emergence is highly variable and dependent on numerous environmental factors.

Steelhead reside in fresh water from one to four years before smolting and out-migrating to the ocean. Juveniles in the Pacific southwest typically spend one to two years before smolting (Barnhart 1986). Steelhead then spends one to four years at sea before returning to spawn. The length of both instream and oceanic residency increases from south to north along the species' distribution (Barnhart 1986).

A.1.3.5 Habitat Requirements

The anadromous and resident forms of rainbow trout are genetically indistinguishable, and habitat requirements of resident rainbow trout are similar to those of steelhead while in the freshwater phase (with the possible exception of estuary and some mainstem habitats).

A.1.3.5.1 Spawning Habitat

Spawning usually occurs in pool tails with cool, clear, well-oxygenated water with suitable current velocity, depth and gravel size (Reiser and Bjornn 1979). Depending on the watershed and size of the fish (resident or anadromous), steelhead spawn at depths of 0.10-1.5 meters, in current velocities of 0.23-1.55 m/sec and in gravel of 0.64-12.7 cm in diameter (Smith 1973; Barnhart 1986). Generally summer run steelhead spawn in the upper sections of watersheds, utilizing habitat inaccessible to fall/winter run fish. Steelhead often utilize intermittent streams for spawning purposes (Kralik and Sowerwine 1977; Carrol 1984).

A.1.3.5.2 Rearing Habitat

After emergence, steelhead fry tend to school and seek out shallow water along stream margins. As the fry grow they start to establish and defend individual territories. Most young-of-the-year steelhead fry inhabit riffles or runs (Barnhart 1986). Mortality of juvenile steelhead is highest the first few months after emergence as fry move about and attempt to establish territories (Shapovalov and Taft 1954; Chapman 1966). Larger steelhead fry (age 1+ year and older) generally maintain territories in pool and run habitats. A productive steelhead stream should have summer temperatures of 10° C to 15° C and an upper limit of around 20° C (Barnhart 1986).

A.1.4 Coastal Cutthroat Trout (*Oncorhynchus clarki clarki*)

A.1.4.1 Listing Status

Coastal cutthroat trout were listed as endangered in the Umpqua ESU in 1996. On April 5, 1999, NMFS determined that listing was not warranted for the Oregon Coast ESU. However, the ESU was designated as a candidate for listing due to concerns over specific risk factors. This ESU included populations of coastal cutthroat trout in Oregon coastal streams south of the Columbia River and north of Cape Blanco (including the Umpqua River Basin. On April 5, 1999, NMFS also determined that listing was not warranted for the Southern Oregon/California Coast Cutthroat trout ESU. The ESU included populations of coastal cutthroat trout from south of Cape Blanco to the southern extent of the subspecies' range (approximately the Mattole River in California). This species is now formally under the jurisdiction of the U.S. Fish and Wildlife Service and at the current time a review of the status of this species in being conducted.

A.1.4.2 Distribution

Coastal cutthroat trout are found in coastal drainages from the Eel River in northern California (Dewitt 1954) to Prince William Sound in Alaska (Trotter 1989). The inland limits of coastal cutthroat trout distribution are most likely the Fraser River in British Columbia and Celilo Falls on the Columbia River (Crawford 1979; Trotter 1989).

A.1.4.3 Life History

The life history of coastal cutthroat trout has been reviewed by numerous authors (Dewitt 1954; Sumner 1962; Armstrong 1971; Johnson 1981; Pauley et al. 1989; Trotter 1989; Behnke 1992). Trotter (1989) described three typical life history forms of coastal cutthroat trout: an anadromous form, a potamodromous form that includes lake and stream-dwelling populations and a non-migratory form which lives in small streams and headwater tributaries. Anadromy tends to be poorly developed. Anadromous populations occur sympatrically and allopatrically with resident populations throughout their distribution (Michael 1989; Pauley et al. 1989; Trotter 1989).

Depending on time of entry, coastal cutthroat trout spawn from December to May. In California, Oregon, Washington and southern British Columbia the peak month is February, whereas in Alaska spawning peaks in April and May. The age of first time spawning females ranges from two to five years old.

Coastal cutthroat trout may spawn more than once. Sumner (1962) reported that in an Oregon coastal stream 39% of coastal cutthroat survived their intial spawning migration, 17% survived a second spawn and 12% survived a third spawn. These data were collected on a watershed lacking an intensive coastal cutthroat fishery.

The fecundity of female coastal cutthroat varies with age and size. Scott and Crossman (1973) reported a range of values from 226 eggs from a 20 cm fish to 4,420 eggs from a 43 cm fish. Forty coastal cutthroat trout collected from McDonald Creek in northern California had an average fecundity of 1,400 eggs (Taylor 1996).

Eggs of coastal cutthroat trout hatch after six to seven weeks of incubation, depending on water temperature. The alevins remain in the gravel approximately two weeks before emergence. The emergence of coastal cutthroat trout fry occurs from March through June, depending on the locale and time of spawning (Trotter 1989).

Anadromous coastal cutthroat trout smolt and migrate to the ocean between the ages of one to six years old (Trotter 1989). There seems to be a relationship between the age and size of smolting and the type of marine environment the smolts enter. For example, in McDonald Creek where smolts enter an enclosed brackish lagoon, a majority of cutthroat smolts out-migrated as one year olds (Taylor 1996). Smolts from coastal watersheds which flow directly into rough surf that forces them offshore tend to out-migrate as three to six year olds. Johnson (1981) speculated that physical and biological characteristics of the marine environment have exterted selective pressures to account for the differences in smolt age and size.

Potamodromous coastal cutthroat trout display migratory patterns similar to anadromous cutthroat, except the resident fish do not migrate to the ocean. Instead their migrations consist of foraging during the spring and summer in main channels of watersheds or in lakes and then migrating into tributaries for spawning purposes (Trotter 1989). Spawning tributaries may be either upstream or downstream from feeding areas. Potamodromous coastal cutthroat trout utilize similar spawning habitat as anadromous forms, and may even contribute to or maintain anadromous populations (Royal 1972; Jones 1979).

Non-migratory coastal cutthroat trout that live in isolated headwater tributaries, remain small in size (150-200 mm), and seldom live past the age of three or four years (Trotter 1989). Females tend to mature by the age of two or three years. The entire life cycle of non-migratory cutthroat trout may be completed in less than 200 meters of stream channel (Wyatt 1959).

A.1.4.4 Habitat Requirements

Coastal cutthroat trout spawn in cool, well oxygenated water with suitable velocity, depth and substrate composition. Coastal cutthroat tend to spawn in first and second order tributaries and isolated headwaters where interactions with other salmonids (primarily steelhead and coho salmon) are reduced (Johnson 1981). Redd sites are generally located in pool tails with protective cover nearby. Spawning has been observed in velocities ranging from 0.11 to 1.02 m/sec, in riffle depths of 0.10 to 1.00 meters and in substrate 0.6 to 10.2 cm in diameter (Smith 1973; Pauley et al. 1989; Taylor 1996).

Total length of newly emerged fry is about 25 mm. They move into low velocity areas along the stream margin, backwater pools and side channels (Moore and Gregory 1988). Fry will remain in these habitats for the entire summer if there is little or no competition from other salmonid species. However, larger coho salmon fry exert social dominance over cutthroat fry and force cutthroat fry into riffles, where they stay until autumn when lower water temperatures reduce aggression in coho and/or elevated flows displace them from the riffles (Glova and Mason 1976).

A.1.5 Tailed Frog (*Ascaphus truei*)

A.1.5.1 Listing Status

This species previously was considered a Category 2 candidate for listing; USFWS subsequently has dropped the “C2” category in its list of species that are listed, proposed for listing, or candidates for listing.

A.1.5.2 Distribution

The tailed frog is the only member of the genus *Ascaphus.* It is endemic to the Pacific Northwest and is widely distributed from northwestern California to British Columbia and western Montana (Nussbaum et al. 1983). Tailed frogs are found at elevations from sea level to near timber line throughout the coastal mountains from British Columbia south to Mendocino County and in the inland mountains of southeast Washington, Idaho, and Montana (Metter 1968). In California, they occur from sea level to 6500 feet, mostly at sites receiving over 40" of precipitation annually in Siskiyou, Del Norte, Trinity, Shasta, Tehama, Humboldt, Mendocino, and possibly Sonoma counties (Bury 1968).

Throughout much of its range the species is distributed as disjunct populations (Metter 1968). Bury and Corn (1988a) believed that isolated, discrete populations most likely occurred in drier forests and heavily managed lands.

A.1.5.3 Life History

Tailed frogs have been shown to breed in both the spring and early fall in different portions of their range. Breeding occurs in the water with the males utilizing the "tail" as a copulatory organ to accomplish internal fertilization. Eggs are deposited in the summer and hatch after four to six weeks (Brown 1990). In coastal regions, the tadpoles typically do not emerge from the nest site until later in the fall (Wallace and Diller, 1998); in interior regions, they over-winter at the nest site and emerge the next spring (Metter 1964). The tadpoles metamorphose into adults in varying time periods depending on the characteristics of the regional population. The larval period may last for 1-4 years; it is shorter in more coastal and lower elevation populations and longer in more inland and higher elevation populations (Daugherty and Sheldon 1982, Nussbaum et al. 1983, Metter 1964 and 1967, Brown 1990, and Wallace and Diller, 1998t).

Adult tailed frogs feed on a wide variety of terrestrial and aquatic invertebrates (Metter 1964). They feed both in the water and on land, and may actively forage in adjacent forests on wet or rainy nights (Nussbaum et al. 1983). The tadpoles feed primarily on diatoms which they scrape off rocks with an enlarged suction-like mouth. Their suction-like mouth also enables them to attach themselves to rocks and other objects in swift flowing water to prevent being washed downstream.

A.1.5.4 Habitat Requirements

Tailed frog habitat has been characterized as perennial, cold, fast flowing mountain streams with dense vegetation cover, or streams in steep-walled valleys in nonforested areas (Bury 1968, Nussbaum et al. 1983). The frogs may inhabit spray drenched cliff walls near waterfalls (Zeiner et al. 1988), but avoid marshes, lakes, and slow sandy streams (Daugherty and Sheldon 1982).

To support larval tailed frogs, streams must have suitable gravel and cobble for attachment sites and diatoms for food (Bury and Corn 1988a). Streams supporting tailed frogs have been found primarily in mature (Bury and Corn 1988a, Welsh 1990) and old growth coniferous forests (Bury 1983, Welsh 1990). Bury and Corn (1988a) reported that the frogs seem to be absent from clearcut areas and managed young forests (Welsh 1990), although they have been observed to occur commonly in young managed forests in coastal California Diller and Wallace, 1999). Welsh (1990) also observed them in young naturally regenerated forests and suggested that structure rather than age per se of old growth was important to the animals. In California, tailed frogs have been found in Sitka spruce, redwood, Douglas-fir, and ponderosa pine forests (Bury 1968).

A.1.6 Southern Torrent Salamander (*Rhyacotriton variegatus*)

A.1.6.1 Listing Status

This species previously was considered a Category 2 candidate for listing; USFWS subsequently dropped the "C2" category in its list of species that are listed, proposed for listing, or candidates for listing. On June 6, 2000 the USFWS announced that, after review, the southern torrent salamander did not warrant listing as endangered or threatened at this time. USFWS recommended that the species remain on the Federal Species of Concern list.

A.1.6.2 Distribution

The southern torrent salamander is one of four species in the genus *Rhyacotriton* and is the most southerly ranging. Recent genetic studies (Good and Wake 1992) split the former Olympic salamander (*R. olympicus*) into four separate species. It is the only species of the genus that occurs in California. Southern torrent salamanders occur west of the Cascades from northwestern Oregon south to Mendocino County in California (Good and Wake 1992). Bury and Corn (1988a) believed that the salamanders are distributed as isolated, discrete populations, especially in heavily managed or drier forests. In California, the species is found in the coastal forests of northwestern California south to Mendocino County (Anderson 1968).

A.1.6.3 Life History

The southern torrent salamander has an aquatic dependent larval stage that may last for two to four years (Nussbaum and Tait 1977) followed by metamorphosis into an adult form. The larvae occupy the interstices among gravels and cobble in the stream. Transformed adults occur in the same microhabitats as the larvae, but are also found under objects along stream edges and in splash zones. Both larvae and adults feed on a variety of small aquatic and semiaquatic invertebrates that are located in the stream or along the margins of the stream (Bury and Martin 1967, Bury 1970). These salamanders are generally believed to have low dispersal capabilities, with annual in-stream movements reported to be usually only several meters (Nussbaum and Tait 1977, Welsh and Lind 1992). However, there is evidence based on pitfall traps that adults can disperse significant distances of up to about 100 meters from streams during wet periods of the year (Grialou et al. 1995).

Breeding is thought to occur for an extended period of time, with the peak of egg-laying probably in spring or early summer (Nussbaum and Tait 1977). Little is known about the selection of sites for egg-laying, but the incubation period is believed to be long, which would result in the eggs over-wintering in the stream.

A.1.6.4 Habitat Requirements

In general, these salamanders occupy humid coastal (Anderson 1968) coniferous forests at maximum elevations that were thought to be 3900 feet (Welsh 1990); but recent field surveys (Diller unpubl. Report) indicate that they can be found up to approximately 5000 feet. They are most commonly associated with the uppermost portions of cold, well shaded permanent streams with a loose gravel substrate (Anderson 1968, Nussbaum et al. 1983), springs, headwater seeps (Welsh 1990), waterfalls (Bury and Corn 1988a), and moss covered rock rubble with flowing water (Anderson 1968). Torrent salamanders also can be found in streams with little surface flow, and they may persist in streams with segments of subsurface flow during the dry summer season. The adult salamanders may inhabit moist stream banks and splash zones, but are rarely found more than 1 m from water (Nussbaum and Tait 1977). They have been observed wintering in talus slopes (Herrington 1988). Bury (1983) did not find torrent salamanders in 6-14 year old logged streams and Bury and Corn (1988a) found the salamanders to be more numerous in streams in uncut 60-500 year old stands than in 14-40 year old regenerated area stands (Bury and Corn 1988a). However, in coastal young growth forests, Diller and Wallace (1996) reported finding no relationship between torrent salamander occurrence and stand age and found salamanders in a high proportion of streams, including recently logged areas.

The other salamander that most closely occupies the same stream microhabitat as the torrent salamander is the larval stage of the Pacific giant salamander (*Dicamptodon tenebrosus*). The Pacific giant larvae grow larger in size and not only compete with torrent salamanders, but probably also prey on them. It is unknown whether Pacific giant salamanders exclude or limit torrent salamanders from certain streams or segments of streams, but have been reported to eat torrent salamander eggs (Nussbaum et al. 1983).

A.2 SENSITIVITY OF THE COVERED SPECIES TO IMPACTS

A.2.1 Anadromous Salmonids

The causes of decline of anadromous salmonids in California are numerous and often interactive but can be grouped into four general categories:

- Degradation or loss of freshwater habitat.
- Interactions with hatchery salmonids.
- Overexploitation of stocks by commercial fishing.
- Climatic factors such as ocean conditions and precipitation timing and amounts.

A.2.1.1 Habitat Degradation and/or Loss

According to Nehlsen et al. (1991) and Reeves and Sedell (1992), degradation and/or loss of freshwater habitat is the single largest cause in the decline of anadromous salmonids along the Pacific northwest Watershed disturbances associated with urbanization, timber harvesting, mining, agriculture, livestock grazing, dams, and water diversions have all contributed to the loss of freshwater habitat.

These human activities have typically reduced the complexity of habitat often associated with productive salmonid streams, especially reductions of LWD and increased sedimentation in pools and spawning riffles (Sandercock 1991). Sedimentation (resulting in shallowing of pools) and removal of riparian vegetation has also lead to excessive increases in summer water temperatures in some salmonid watersheds.

Loss of spawning and rearing habitat has also occurred through human activities which denied migrating adults access to traditional spawning areas. Dams on the Klamath, Trinity, Mad, Eel, Sacramento and San Joaquin Rivers have all severely impacted runs of salmon and steelhead in California. These dams have either prohibited fish access to traditional spawning and rearing areas and/or degraded downstream habitat conditions. Improperly installed culverts have reduced or prohibited access of migrating spawners to tributaries within numerous coastal watersheds.

A.2.1.2 Interactions with Hatchery Salmonids

Interactions with hatchery salmonids have possibly impacted wild stocks of salmonids through:

- potential loss of genetic integrity;
- competition between juveniles;
- transmission of diseases.

Although widely cited as occurring, the loss of genetic integrity is difficult to determine because the amount of interbreeding between native and non-native stocks is poorly understood (Hindar et al. 1991). Stocks of coho in California do not appear to be strongly differentiated genetically (Bartley et al. 1992). This lack of differentiation may be caused by transplants of stocks within California plus the introduction of coho from various Oregon and Washington watersheds decades prior to the ability to determine an individual's genetic composition (Bartley et al. 1992).

Several studies have reported reduced densities of wild juvenile coho after the release of hatchery juveniles (Nickelson et al. 1986; Miller et al. 1990). Miller et al. (1990) also reported similar reductions in the subsequent adult returns. In subsequent years, Nickelson et al. (1986) detected a shift towards earlier returning adult spawners, which is indicative of hatchery fish (Brown et al. 1994). These reductions in native juvenile densities may occur because juvenile coho are territorial and the larger hatchery fish displace the natives from preferred habitat (Nickelson et al. 1986). When displaced from established territories, juvenile coho are more suceptible to predation and may also experience reduced growth rates which may further affect survival to maturity (Puckett and Dill 1985; Steward and Bjornn 1990).

The transmission of diseases from hatchery salmonids to native stocks is potentially a serious problem, yet little information exists to confirm the extent of this concern because of limited field investigations (Steward and Bjornn 1990; Kruger and May 1991). An example of hatchery salmonids passing diseases to wild fish recently occurred in the Madison River in Montana where planted rainbow trout infected the wild population with whirling disease. In three years the Madison River's rainbow trout population declined by more than 90% (Holt 1995). The following virulent diseases affect hatchery salmonids and have the potential to infect wild stocks: viral hemorrhagic septicemia, bacterial kidney disease, infectious hematopoietic necrosis, herpes virus and infectious pancreatic necrosis (Håstein and Lindstad 1991).

A.2.1.3 Over-exploitation

Excessive harvest by commercial fishing is commonly cited as a significant factor in the decline of chinook and coho salmon, but the effects are hard to quantify since catch records rarely distinguish between wild and hatchery stocks (Steward and Bjornn 1990). In mixed-stock commerical fisheries, wild stocks may be overfished because they are unable to sustain the same harvest rates as hatchery fish.

Female coho salmon in California mainly have a three year life cycle, thus they lack the ability to withstand overharvest compared to other salmonids in which a single year class matures at a variety of ages. For example, the coho runs in Scott and Wadell Creeks (the southern most coho populations) have exclusively three year life cycles and only experience a strong return once every three years because two of the year classes are severely depressed (Brown et al. 1994).

Although steelhead are not fished commercially in the United States, exploitation by foreign fleets has been blamed in the decline of steelhead stocks. Asian fleets gillnetting squid in the Gulf of Alaska have been long suspected as a major harvester of steelhead from North American watersheds.

In-river gillnetting by native American tribes has also been suggested in the decline of some salmonid stocks. While these fisheries are currently regulated to allow sufficient escapement of adults, regulations concerning the timing and gear restrictions of these fisheries may impact certain segments of salmon runs. For example, timing of the fishery may over-harvest an early or late segment of a run. On the Klamath River, regulations require large gillnet mesh sizes to prevent the harvest of steelhead. However, large mesh sizes target larger chinook salmon and may have contributed to the decline of older age classes of spawning adults.

A.2.1.4 Climatic Factors

Although extremely difficult to quantify, recent natural climatic events have most likely contributed to the decline of numerous stocks of anadromous salmonids along the Pacific northwest coast. A warming trend in the ocean along the Pacific northwest coast during 1976-1983 coincided with: 1) an abrupt drop in coho adult numbers in the Oregon Production Zone; 2) elevated sea-surface temperatures; and 3) reduced biological productivity in the California Current (Nickelson 1986; Lawson 1993). The 1982-1983 El Niño event, the largest ocean warming event of the century, severly impacted primary and secondary productivity thus impacting the entire northeast Pacific food-web (Pearcy 1992).

California is the southernmost range of coho salmon and these populations are well adapted to the extreme hydrologic, physical and climatic conditions (for salmonids) of coastal watersheds. However, the recent drought conditions of 1976-177 and 1986-1992 have made survival of the species in the southern part of its range even more demanding. Instream salmonid habitat conditions during the droughts were impaired by the sucessive years of low rainfall.

Conversely, past flood events have also impaired coho salmon habitat along the Pacific northwest coast. The recent floods of 1955 and 1964, in combination with intensive pre-Forest Practice Rules timber harvesting, severely degraded the quantity and quality of salmonid habitat in northern California watersheds. Salmonids in California have certainly experienced catastrophic natural events in the distant past, but these past salmonid populations were not simultaneously confronted with widespread, continuous human-related impacts to instream habitat.

A.2.2 Tailed Frog

Tailed frogs were considered rare for many years, but are now known to occur in high densities in suitable habitats (Nussbaum et al. 1983). Welsh (1990) expected numbers of frogs to decline due to timber harvest, to which they seem sensitive (Bury and Corn

1988b). He also speculated that the narrow niche, isolated population distribution, and long generation time of tailed frogs in conjunction with the lack of protection of headwater habitats make the species susceptible to local extirpations. Bury and Corn (1988a) predicted that populations subjected to clearcutting in the Coast Range of Oregon and northern California would probably go extinct following clearcutting, whereas those in the Cascades of Oregon and Washington had a higher probability of surviving. However, Bury (1968) noted that deforestation had a less detrimental effect on tailed frog populations where an influence of maritime climate was present. Studies in the coastal areas of northern California (Diller and Wallace, 1999) support the hypothesis that the impacts of timber harvest are less in coastal areas. Similar too what was noted above for the torrent salamander, tailed frogs were found in a high proportion of streams in previously logged areas. Geology was also the most important landscape-scale variable associated with occurrence of tailed frogs.

Bury and Corn (1988a) and Welsh (1990) believed that long-term, range-wide reductions or extinctions of tailed frogs were likely due to local extirpations, increased population fragmentation, habitat loss, restricted gene flow, and limited recolonization of streams when habitats are re-established (Bury and Corn 1988a).

Removal of timber by logging or fire is believed to result in the disappearance of tailed frogs due to increased stream temperatures (Noble and Putnam 1931, Nussbaum et al. 1983, Bury and Corn 1988a) and sedimentation (Nussbaum et al. 1983, Bury and Corn 1988a). The effects may affect the frogs directly, or indirectly by shifting the larval food base from diatoms to green algae (Bury and Corn 1988a). However, Bury (1968) stated "Presence of the frog in logged areas of coastal Humboldt County suggests that deforestation is less of a threat to the disappearance of *Ascaphus* in coastal than inland streams".

Although the survival of tailed frogs may depend on protection of cool flowing streams and adjacent forest habitats (Bury and Corn 1988b), timber harvest is not incompatible with such protection (Welsh 1990). Bury and Corn (1988a) and Welsh (1990) suggested eliminating harvest adjacent to aquatic habitats and establishing buffer strips to protect current frog populations and act as sources for future repopulation of logged areas. Bury and Corn (1988a) recommended establishing protection zones by retaining deciduous and small (cull) trees around streams while felling merchantable timber away from the streams. They noted that small clumps of trees around streams rather than cover along whole stream courses may be adequate to protect local populations (Bury and Corn 1988a). Retention of coarse woody debris for nutrient sources and sediment traps, further studies and surveys of tailed frogs, and protection of headwater habitats have also been recommended (Bury and Corn 1988a).

A.2.3 Southern Torrent Salamander

Welsh (1990) believed that logging and fragmentation of old growth coniferous forests would cause numbers of torrent salamanders to decline, with local extirpation of populations due to the species microhabitat requirements and lack of protection of headwater habitats. Bury and Corn (1988a) suggested that recolonization of logged areas would be rare and slow due to isolated population distribution, long generation time, narrow temperature requirements, and susceptibility to water loss limiting overland dispersal of the species (Welsh 1990). Recolonization may be more likely to occur in high gradient streams (Bury and Corn 1988a), but Welsh (1990) thought that local extirpations, increased population fragmentation and habitat loss, and restricted gene flow made populations vulnerable to long-term range-wide extinctions. The impacts of timber harvest on torrent salamanders appear to be less severe in coastal areas. Diller and Wallace (1996) found a high proportion of salamanders in streams that previously had been logged, including recently clearcut areas. In these coastal areas, geology was the only landscape-scale variable that strongly correlated with the occurrence of salamanders. In areas of a consolidated geologic type (e.g., Franciscan), torrent salamanders were found in high gradient reaches of almost all streams that were searched. It was hypothesized that the cool moist conditions of the coastal areas ameliorate the impacts of canopy removal for this species.

Short-term detrimental effects of logging on salamander habitat include increased sedimentation which fills crevices, and increased water temperatures (Bury and Corn 1988a). Bury and Corn (1988b) noted that these salamanders were sensitive to timber harvest and suggested that their survival was dependent on the protection of cool flowing streams and adjacent forested habitats which provide shade and maintain stream quality. Timber harvest plans should be designed and implemented to provide such protection (Welsh 1990). Bury and Corn (1988a) recommended protecting streams by felling merchantable timber away from streams and leaving deciduous and small (cull) trees to provide shade cover. To reduce the expense of leaving merchantable timber along whole stream courses, small clumps of trees may be retained to protect current populations and provide sources for future repopulation of logged areas (Bury and Corn 1988a). Retaining coarse woody debris, conducting preharvest surveys, and obtaining more data on the species' habitat preferences and environmental tolerance have also been recommended (Bury and Corn 1988a).

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